EVALUATION OF TERRESTRIAL AND STREAMSIDE SALAMANDER MONITORING TECHNIQUES AT SHENANDOAH NATIONAL PARK

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Abstract. In response to concerns about amphibian declines, a study evaluating and validating amphibian monitoring techniques was initiated in Shenandoah and Big Bend National Parks in the spring of 1998. We evaluate precision, bias, and efficiency of several sampling methods for terrestrial and streamside salamanders in Shenandoah National Park and assess salamander abundance in relation to environmental variables, notably soil and water pH. Terrestrial salamanders, primarily redback salamanders (*Plethodon cinereus*), were sampled by searching under cover objects during the day in square plots (10 to 35 m^2). We compared population indices (mean daily and total counts) with adjusted population estimates from capture-recapture. Analyses suggested that the proportion of salamanders detected (p) during sampling varied among plots, necessitating the use of adjusted population estimates. However, adjusted population estimates were less precise than population indices, and may not be efficient in relating salamander populations to environmental variables. In future sampling, strategic use of capture-recapture to verify consistency of p's among sites may be a reasonable compromise between the possibility of bias in estimation of population size and deficiencies due to inefficiency associated with the estimation of p. The streamside two-lined salamander (*Eurycea bislineata*) was surveyed using four methods: leaf litter refugia bags, 1 m^2 quadrats, 50×1 m visual encounter transects, and electric shocking. Comparison of survey methods at nine streams revealed congruent patterns of abundance among sites, suggesting that relative bias among the methods is similar, and that choice of survey method should be based on precision and logistical efficiency. Redback and two-lined salamander abundance were not significantly related to soil or water pH, respectively.

1. Amphibian Investigations in the Mid-Atlantic Integrated Assessment Region

Early in the conceptualization of the Mid-Atlantic Integrated Assessment (MAIA), it was recognized that biodiversity would be a key indicator and integrator of changing environmental factors and resources. Amphibians were selected as one of the classes for study because of increasing reports of amphibian population declines and malformations, their significant role in ecosystems, and their sensitivity to pollutants and other environmental stressors.

In addition to considerations regarding the research goals of MAIA, momentum in the EPA was increasing for the development of biocriteria as part of the Agency's water monitoring objectives. In the early 1990s, the EPA began to integrate biological survey methods alongside toxicity testing and measurements of chemical and physical parameters for assessing the ecological integrity of aquatic ecosystems (USEPA 1990). Because MAIA scientists were charged with develop-

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ing scientifically valid tools to support the EPA's new initiatives in establishing criteria for the protection of aquatic life, research into amphibian monitoring methods seemed appropriate.

Unfortunately, baseline historical data on amphibian populations in the MAIA region, against which current populations can be compared, are scarce and evidence for amphibian declines in the MAIA region is often anecdotal. Reliable methods for sampling amphibian populations and validation of relative abundance indices is fundamental to the creation of rigorous population monitoring programs.

To address these issues, MAIA has sponsored several efforts through an Interagency Agreement with the USGS Biological Resources Division. First, an "Amphibian Information Website" was developed to support amphibian conservation efforts in the mid-Atlantic (*http://www.mp2-pwrc.usgs.gov/amphibs*), in which bibliographic searches can be submitted using keywords, geographic regions, and taxonomy. Second, over 200 published counts of amphibian populations were used to evaluate the statistical power of amphibian monitoring techniques for eastern North American amphibians. The program "MONITOR" (*http://www.mp1-pwrc.usgs.gov/powcase/powcase.html*) was used to investigate the relationship between survey designs and the power to detect amphibian population trends over time. The website provides a stepwise approach to determining adequate sample sizes, which take data precision and variability into account. Wildlife managers can use these websites to develop efficient and valid amphibian monitoring programs.

Several efforts have been implemented in the MAIA to evaluate salamander population monitoring techniques and how salamanders may serve as bioindicators of environmental stressors (e.g., acid precipitation and deposition, sedimentation, and land use change) in aquatic and terrestrial environments (G. Rocco, personal communication). First, researchers at Pennsylvania State University are surveying streamside salamanders in forested headwater streams, which comprise 60–75% of the total stream length and watershed area in the Mid-Atlantic states. Because of their diverse and complex life histories, abundant, stable, and geographically widespread populations, stream salamanders provide another ecological tool to assess headwater habitat quality where other species assemblages may be poorly developed or absent. Second, a Mid-Atlantic anuran calling survey is being funded by the Regional Environmental Monitoring and Assessment Program (REMAP) through a collaborative effort among the five MAIA state natural resource programs. This calling survey is to be modeled after the Wisconsin Frog and Toad Survey established in 1984. Third, Park Research and Intensive Monitoring of Ecosystems Network (PRIMENet), is supporting amphibian monitoring and research at several National Parks, including the Shenandoah National Park research described here.

In addition to these studies, herpetofaunal surveys have recently been incorporated into state survey programs (Maryland Department of Natural Resources

1995) within the MAIA region. Also, at least one state is considering monitoring streamside salamanders in small headwater streams where they replace fish as the dominant predator (R. Davic, personal communication).

2. Salamander Monitoring in Shenandoah National Park

2.1 IMPORTANCE OF SALAMANDER MONITORING

Amphibian declines world-wide have created a need for more extensive and standardized monitoring of amphibian populations and for elucidating underlying causes of amphibian declines (Heyer et al. 1994). Monitoring programs yield data on the geographic extent and magnitude of population changes and allow researchers to link population changes with possible environmental and anthropogenic factors. Salamanders might be particularly susceptible to declines. Compared to many anurans, salamanders are relatively long-lived, take longer to reach maturity, and lay fewer eggs (Petranka 1998). Salamanders constitute a large biomass in eastern deciduous forests and streams and are important in ecosystem energy flow and nutrient cycling (Burton and Likens 1975, Wyman 1998). Hence, changes in salamander populations could impact predator and prey populations and ecosystem processes. Salamanders are sensitive to environmental factors such as acidification, drought, contaminants in soil and water, and habitat destruction or alteration (Wyman and Hawksley-Lescault 1987, Storm et al. 1994, deMaynadier and Hunter 1995, Ash 1997). In Shenandoah National Park, located in the Blue Ridge Mountains of Virginia, the primary threats to salamanders have been: 1) habitat alteration (e.g., past land use including logging, agriculture, and road and trail construction, and tree defoliation and mortality caused by gypsy moths), and 2) acidification of soils and water (Mitchell 1998). In this paper, we investigate the ability of salamander monitoring techniques to provide unbiased and precise measures of abundance, and analyze the relationship between salamander abundance and soil and water pH.

2.2 SALAMANDER SPECIES AT SHENANDOAH NATIONAL PARK

Terrestrial salamanders in Shenandoah National Park include the redback salamander (*Plethodon cinereus*), white-spotted slimy salamander (*P. cylindraceus*), and the federally endangered Shenandoah salamander (*P. shenandoah*) (Witt 1993). Plethodontids are lungless salamanders, relying on moist skin as a respiratory surface. During the day, these salamanders are found under cover objects such as logs and rocks during moist conditions (high humidity, recent rains) or below ground within moist retreats during dry conditions. On wet nights when temperatures are above freezing, terrestrial salamanders can be found foraging above ground. Streamside salamanders in Shenandoah National Park include the northern dusky salamander (*Desmognathus fuscus*), seal salamander (*D. monticola jeffersoni*), northern spring salamander (*Gyrinophilus porphyriticus*), northern red salamander (*Pseudotriton ruber*), and two-lined salamander (*Eurycea bislineata*) (Witt 1993). These salamanders are found in and alongside streams, seeps, and springs, under rocks and other objects.

2.3 ISSUES IN SALAMANDER MONITORING

Gathering baseline salamander population data is essential for evaluating population changes over time, and for assessing relationships between salamander populations and environmental factors. Salamanders tend to be difficult to monitor because of their seasonal and weather-dependent occurrence, fossorial habits, cryptic nature, and lack of vocalization.

Another problem with monitoring salamanders, and amphibians in general, is whether one can or should statistically estimate undercounts (i.e., the proportion of the population not counted during a survey because they are underground, not calling, missed by the observers, or otherwise unobservable). Population sampling techniques can be categorized as: 1) population indices, based on the numbers of individuals counted in an area, or 2) adjusted population estimates, which estimate the "true" population based on capture-recapture or other methods. Population indices, such as the number of salamanders found under logs or along transects on rainy nights, are easy to collect, yet count only a portion of the actual population present. On the other hand, adjusted population estimates are labor-intensive but estimate the undercount in sampling and may yield a more accurate picture of the number of salamanders present. Adjusted estimates of salamander populations have been difficult to obtain because of inadequate marking techniques and low recapture probabilities (Taub 1961, Banasiak 1974, Gergits and Jaeger 1990). A further complicating factor is that salamander capture probabilities are influenced by a number of factors, including site-specific characteristics (e.g., cover object density), weather conditions, and time of day.

Sampling methods and indices must be evaluated in terms of their bias, precision, and logistical efficiency. Bias is the "difference between the expected value of a population estimate and the true population size" (Lancia et al. 1994). Generally, population indices are biased as estimates of total population size, but adjusted population estimates have no (or little) bias because they estimate a detection rate (p) as part of the procedure. If estimation of population size is needed, researchers should preferentially choose sampling methods that are not biased. If population change over space or time is the parameter of interest, indices can be used for the estimation if their bias is consistent over space and time. Precision is a "measure of how close a population estimate is to its expected value" (Lancia et al. 1994). The most useful sampling method or population index is one which is both unbiased and shows the least variation or measurement error. One is more likely to find significant differences among sites or treatments or between population estimates and environmental variables if the population index is less variable. Logistical efficiency, the "capacity to produce desired results with a minimum expenditure of energy, time, money, or materials" (Webster's 3rd New International Dictionary 1968), is also a consideration in survey optimization. Bias, precision, and logistical efficiency of sampling strategies have important consequences for species conservation and for documenting associations between populations and environmental factors. Understanding relationships between environmental factors and amphibian populations are critical for modeling and managing populations. The use of population indices without adjustments for undercounts may produce inaccurate views of these relationships.

2.4 COMPARATIVE APPROACHES TO ESTIMATING SALAMANDER POPULATIONS AND TRENDS

In this paper, two comparative approaches are taken to determine bias and precision associated with population sampling methods for terrestrial and streamside salamanders. For terrestrial salamanders, bias in alternative estimates was evaluated by estimating the proportion of salamanders detected (p) using capture-recapture and testing whether the p's differed among plots. If p's differ among plots, the population indices based on the number of animals captured yields biased estimates of abundance, and correction factors must either be applied to the indices, or population estimation techniques must be used before accurate comparisons can be made of salamander population sizes among localities. Bias in indices may also affect estimation of population changes over time if factors that can lead to bias—such as changes in leaf litter depth or the number of cover objects—change greatly over time. Alternatively, if p's do not differ among plots, population indices will provide unbiased estimates and may be used directly to compare abundance among plots (Skalski and Robson 1992).

For streamside salamanders, direct estimates of p were not available. Instead, we indirectly assessed the validity of the population indices by determining whether four survey methods yielded consistent patterns of two-lined salamander abundance among nine stream sites. If the four methods yielded different results on abundance at the streams, this would indicate that at least one of the methods was biased, possibly invalidating comparisons based on the index. If the methods yielded similar results, we could infer that they provide consistent abundance data among sites, validating the index assumption.

Precision can be assessed in several ways. One way is to look at the coefficient of variation (CV = standard deviation/mean) in population index counts over time. Higher CVs indicate a less precise or more variable count index. Counts of amphibians tend to have large CVs (S. Droege, unpublished data); for salamanders, CVs within one year above 0.6 (60%) can be considered high, though CVs are primarily assessed in a relative framework (e.g., method A has a higher CV than method B, so method B is preferred). Another way to assess precision is to compare the indices with environmental variables of interest to determine which index yields the most precise relationship to that variable, as indicated by the smallest root mean squared error [RMSE] term around a mean or regression line.

3. Terrestrial Salamander Monitoring at Shenandoah National Park

3.1 METHODS

Various monitoring techniques for terrestrial salamanders have been employed in the past, including area- (e.g., quadrat, transect) and time-constrained searches by day or night, drift fence and pitfall trapping, and artificial cover objects. We searched for salamanders during the day under natural cover objects in square plots as this was the simplest and safest technique to use and has previously been proven effective. Artificial cover object arrays are also used in the Park, but are not discussed here.

In the spring of 1998, six paired plots (15, 20, and 35 m^2) were established in an area near the Park headquarters scheduled for a prescribed burn in the spring of 1999, and in a control area of similar forest type nearby. The plots were visited during the day on four occasions between 8 April and 21 May. One team searched a burn plot while the other team searched the paired control plot simultaneously. All natural cover objects on the surface of the forest floor that could be lifted were checked, and salamanders found underneath were captured and measured for snout-vent length and total length.

Salamanders over 38 mm in total length were marked individually using visible implant fluorescent elastomer (VIE) (Northwest Marine Technologies, Inc.). VIE marking was initially developed for fish (Bonneau et al. 1995), but has recently been used to mark amphibians (*http://www.mp1-pwrc.usgs.gov/marking/vie.html*). The VIE is injected underneath the skin on the right and/or left side of the salamander behind the forelimbs or in front of the hindlimbs. The mark is enhanced by viewing under an ultraviolet light. Using three colors (red, orange, yellow) (C) and four body locations (L) allows for 255 individual marks ($\{[C + 1]^L\} - 1$).

From these data, three measures of population size can be derived: 1) average number of captures per day (mean daily captures), 2) total number of distinct captures over the four trapping occasions (total captures), and 3) estimated population size using capture-recapture. Program Capture (Otis et al. 1978, Rexstad and Burnham 1991), which assumes a closed population (no births, deaths, immigration, emigration), was used to fit one of a series of alternative statistical models to the data at each plot. Although the recapture period in this study encompassed a six-week period, tests for population closure at each plot did not reject the null hypothesis of a closed population (all $P > 0.05$). The Otis et al. (1978) models differ in their assumptions about consistency in detection of animals over time, allowing for heterogeneity over time, or individual behavioral responses to trapping. Chi-square tests are provided to allow investigators to evaluate which model best fits the data. Population estimates and detection rates were estimated for each site, and a chi-square test (Program Contrast, Sauer and Williams 1989) was used to test the null hypothesis that detection rates did not differ among sites.

Environmental variables collected in each plot included the number of overturned rock and wood cover objects, air and soil temperature, soil moisture and pH (University of Delaware 1991), leaf litter depth, percent canopy cover and understory vegetative cover below 1 m, and predominant tree species in the plot. In this paper, we present the relationship between redback salamander abundance and soil pH.

3.2 RESULTS

Capture-recapture models that best fit data from the control and burn plots are presented in Table I. Although low numbers of recaptures made model-fitting problematic for some plots (Menkens and Anderson 1992), the model selection procedure in Program Capture indicated that Models B (behavioral response in capture rates), H (individual heterogeneity response), TH (time and heterogeneity response) and O (no effects on capture rates) were appropriate. The p's estimated from these analyses differed among plots (n = 10; no recaptures at plots C4 or B2 precluded analyses at these plots) ($\chi^2 = 46.2$, $df = 9$, $P < 0.0001$), suggesting that salamander visibility differs among plots.

Model selection can play a role in determining whether p's differ among sites. Model B best fit the data for plots C5, C6, and B1, which had the highest p values. Models that best fit the other plot data sets produced smaller p's. An alternative approach to model fitting would be to use the most commonly selected or dominant model which best fits a majority of the plots (in this case, Model O). Detection probabilities did not differ among sites when we used Model O-derived p values ($\chi^2 = 9.7$, $df = 9$, $P = 0.37$) (Table I). The conclusion that detection rates differed among plots must be viewed as tentative.

In assessing the precision of counts, a different picture emerges. Redback salamander population densities were determined by dividing the counts and estimates by the area of the plot. Densities were less variable using the two population indices (mean daily captures, total captures) compared to the adjusted population estimate N (Figure 1). Plot CVs (calculated using plot means) were similar for the mean daily captures (51%), total captures (46%), and visibility-adjusted population estimates (47%). However, the variances differed greatly, with the estimated N showing the highest variance (0.08), followed by the total population (0.005) and the mean daily indices (0.0006). Of course, among-site variances incorporate both between-site differences in populations as well as measurement error. For the mean daily captures and the adjusted population estimates, the pooled within-site measurement error CVs are 42% and 40%, respectively.

Table I
Capture-recapture estimates for redback salamanders at burn (B) and control (C) plots at Shenandoah National Park.

Plot	Size (m^2)	# Marked	# Recaptured	% Recapture	*N*	SE (*N*)	Daily Proportion Detected (*p*)	Best Model	Model O Daily Proportion Detected (*p*)
C1	35	138	12	9	646	171	0.06	TH	0.06
C2	20	73	8	11	165	16	0.12	H	0.08
C3	15	42	3	7	199	94	0.06	O	0.06
C4	20	34	0	0	-	-	-	-	-
C5	15	33	1	3	42	9	0.31	B	0.03
C6	15	68	2	3	99	21	0.25	B	0.03
B1	35	170	7	4	279	52	0.21	B	0.03
B2	15	37	0	0	-	-	-	-	-
B3	15	63	8	13	220	67	0.08	O	0.08
B4	15	42	3	7	199	94	0.06	O	0.06
B5	20	33	1	3	301	241	0.03	O	0.03
B6	20	26	1	4	282	288	0.03	TH	0.03

N = adjusted population estimate
SE = standard error
Model O = all individuals are equally likely to be caught on every sampling occasion
Model B = individual capture varies by behavioral response to capture
Model H = capture probability varies by individual
Model TH = capture varies with time and with behavioral response to capture

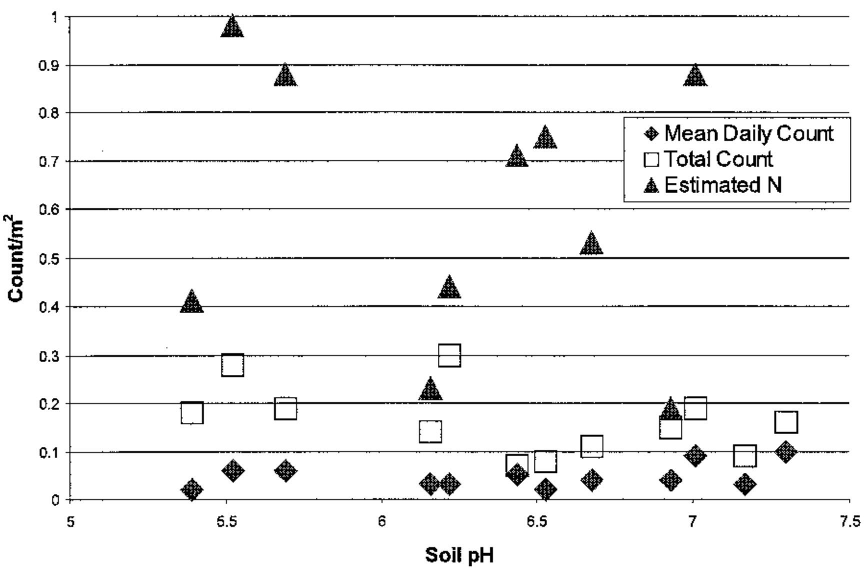

Figure 1. *Population Estimates and Soil pH.*

None of the redback salamander population indices was significantly related to soil pH at the plots ($r^2 < 0.21$, $P > 0.1$). Wyman and Hawksley-Lescault (1987) found significantly fewer redback salamanders in more acid soils, although the soil pH range in their study was lower (pH 2.7 to 5.9). The mean daily and total count RMSE terms around the regression lines were much lower (0.02 and 0.07, respectively) than the RMSE around the adjusted population estimate (RMSE = 0.29) (Figure 1). The RMSE estimates suggest that the count indices would provide more precise views of the relationship between salamander abundance and pH, if bias in the indices is not confounded with among-site differences in abundance.

4. Streamside Salamander Monitoring at Shenandoah National Park

4.1 METHODS

Various monitoring techniques exist for streamside salamanders, including area- (quadrat, transect) and/or time-constrained searches, drift fence and pitfall trapping, and dipnetting. From 4 June to 28 August 1998, nine streams throughout the Park were surveyed for salamanders using four techniques: 1 m^2 quadrats, leaf litter refugia bags, 50 × 1 m visual encounter transects, and electric shocking devices. Transects for the 1 m^2 quadrat, leaf litter bag, and 50 × 1 m surveys were adjacent to the electric shocking transects. Two transects were surveyed at each stream, except at Dry Run, where only one transect was surveyed. We recorded snout-vent length, total length, mass, and age class (larva, adult) for captured salamanders.

One m^2 quadrat searches were conducted as described in Mitchell (1998). A 1 m^2 square made of PVC pipe was placed adjacent to the water's edge every 5 m on both sides of the stream along 100 m transects. To search for salamanders in the quadrat, all rocks in the quadrat were overturned, and the area was further searched by raking with a trowel through the remaining rocks.

Leaf litter refugia bags, as described in Pauley (1995), were made of plastic netting (Deer Block, 7′ × 100′ roll, 1.5 cm^2 mesh) cut into 50 cm^2 squares. Approximately 0.2 kg of small rocks, leaf litter, and moss were placed in the center of the net squares, and a cable tie was woven through the netting at the top and cinched to make a bag. Flagging tape was tied to the top of the bags to make them visible in the stream. Bags were placed every 5 m along both sides of the stream along a 100 m transect and were surrounded and topped by rocks in order to keep them in place. Leaf litter bags were first checked two weeks after placement. To check the leaf litter bag, we immediately placed a dipnet underneath the bag and shook the bag in the dipnet for 10 seconds. We also shook the leaf litter bag in a bucket of water for 10 seconds, and poured the water through the dipnet.

Visual encounter 50 × 1 m shoreline transects were conducted by turning over rocks and logs along both sides of the stream. The search time, number of overturned objects, and number of salamanders observed were recorded.

Electric shocking fish crews from Shenandoah National Park conduct a three-pass removal system to estimate fish populations at streams throughout the Park (Atkinson 1994). A two-person amphibian crew waded behind the fish crew with dipnets to catch any salamanders which surfaced or were observed in the stream.

Environmental variables collected included water quality variables (temperature, pH, conductivity), water depth, channel type (e.g., pool, riffle), stream bank composition (e.g., wood debris, gravel, boulder), and percent canopy cover.

We attempted to obtain adjusted population estimates for streamside salamanders using two techniques. During electric shocking, the three-pass removal sampling allowed for the calculation of an adjusted population estimate (Otis et al. 1978). We also used VIE marking in a capture-recapture study along a 50 × 1 m transect. The stream transect was visited seven times between 28 September and 26 October 1998, and all captured adult salamanders longer than 38 mm were marked.

As with the terrestrial salamander analysis, we were interested in assessing: 1) whether the alternative indices provided consistent abundance data among sites, and 2) which procedure provided the most precise population estimates for comparison with environmental covariables. Because we could not directly estimate bias associated with population estimation at sites for most methods, we assessed bias indirectly using a two-way ANOVA with interaction. If the indices provided consistent measures of abundance among sites (i.e., the method × site interaction was not significant), then the bias in the indices was not changing among sites (or was consistently changing for all indices), validating their use in association analyses. If this was the case, selection of the best index could be based on precision and other considerations.

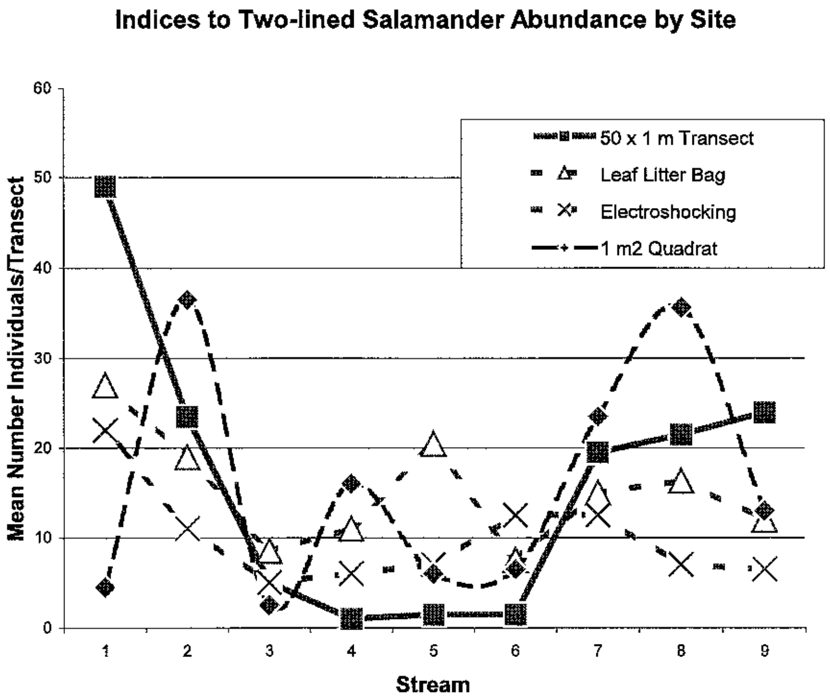

Figure 2. Indices to Two-lined Salamander Abundance by Site.

4.2 RESULTS

We found seven salamander and five anuran species at the nine streams. The most abundant salamander at all sites was the two-lined salamander, so we used this species for subsequent analyses.

We found a significant difference among sites in two-lined salamander abundance (two-way ANOVA: $F_{(8,34)} = 2.26, P = 0.047$) (Figure 2). Sites 3–6 had lower two-lined salamander abundance than the other sites. We found no significant difference among the four survey methods ($F_{(3,34)} = 0.76, P = 0.52$), and the site × survey method interaction term was not significant ($F_{(24,34)} = 1.04$, $P = 0.45$), indicating that across sites, the survey methods showed similar patterns or bias of two-lined salamander abundance.

Two-lined salamander abundance (mean number/transect) did not differ significantly among sites in relation to pH using an ANCOVA with water pH as a covariate (Figure 3). None of the correlations between two-lined salamander abundance using each of the survey methods and pH was significant (all $r^2 < 0.25$, $df = 7, P > 0.18$) , and the method × pH interaction term was not significant. The electric shocking and leaf litter bag methods showed the lowest RMSE around the regressions, indicating that these methods are less variable (more precise) in their relationship to pH. In summary, the survey methods were consistent in their relationships over sites and with pH, indicating consistent bias among sites. However, indices differed in RMSE, with electric shocking showing the most precise relationship with pH. However, of all the methods, electric shocking is the least effec-

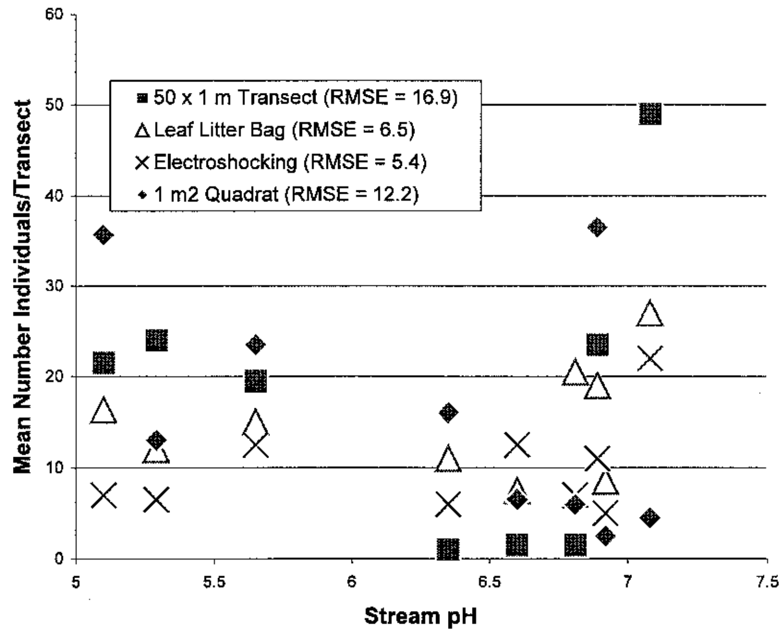

Figure 3. *Indices to Two-lined Salamander Abundance and Stream pH.*

tive in terms of catching salamanders and the least efficient because it is expensive and labor-intensive. Electric shocking may also negatively impact amphibian and fish populations (Roach 1996).

We attempted to assess bias from electric shocking data. Program Capture includes a "removal method" estimate of population size, based on the assumption that removal of animals from the population is similar to a behavioral response to capture. For effective population estimation using removal methods, the numbers of individuals caught should decrease with each capture occasion. This was rarely the case for amphibians during successive electric shocking runs. The number of amphibians caught in successive passes only decreased in 31% ($^5/_{16}$) of the electric shocking transects. Because of this, population estimates either could not be calculated or were very imprecise, and are not presented here.

Salamander recapture rates in the 50 × 1 m stream transect were extremely low. Only one salamander was recaptured out of 38 marked salamanders, and this was a redback salamander. The CV for total counts summing all species was 69%; for individual species, CVs ranged from 75% to 265%. The use of capture-recapture for stream salamanders using the transect technique is not promising. However, these data represent only one site, and dry conditions during the fall may have reduced recapture rates.

5. Implications of Results and Recommendations

Assessment of bias, precision (in terms of measurement error and RMSE), and logistical efficiency is needed in the development of monitoring programs. For redback salamanders, differences in p among sites indicates that capture-recapture methods are needed to evaluate whether bias is changing over sites. Additional work is needed to better estimate and document spatial variation in p. For two-lined salamanders, comparative analyses of indices indicated consistency in bias among sites, but differences in RMSE, suggesting differences in the ability of the indices to test hypotheses of environmental association.

Generally, salamanders have not been monitored using statistical procedures that allow explicit estimation of p. Hence, little is known about spatial and temporal variation in p associated with sampling procedures. In future sampling, capture-recapture will be used at a subset of sites to verify consistency of p's over time and space. Once understanding of temporal and spatial variation of p is achieved, the information can be used to adjust indices. We hope to increase recapture rates by conducting diurnal searches only when certain weather conditions are met, by sampling on wet nights (Ash 1997), and by using artificial cover objects. For streamside salamanders, we will continue to test capture-recapture using leaf litter bag and 50 × 1 m transect methods, and will also try removal sampling within one- and two-day periods.

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